Locations sampled for the first time during Phase 2 included background samples BKSW2, BKSW3, and BKSW4. Each of these samples was analyzed for TCL SVOCs. None of these compounds were detected in these Phase 2 samples.

2.15.3 Pesticides and PCBs in Surface Water

Endrine ketone was the only pesticide detected in surface water samples collected during Phases 1A and 1B of the RI. Endrine ketone was detected in three samples, one background sample (01BKSW1AA (0.021 µg/L)), and two study area samples (01SW16AA (0.021 µg/L)), and 01SW35AA (0.025 µg/L)). None of the reported pesticide results exceeded their respective HHAWQC screening values. PCBs were not detected in any surface water samples collected from the project site during Phase 1A or 1B.

Based on the limited detection of these compounds in surface water samples collected during Phase 1, they were eliminated as an analyte for any surface water samples collected from previously sampled locations during Phase 2 of the RI.
Locations sampled for the first time during Phase 2 included background samples BKSW2, BKSW3, and BKSW4. Each of these samples was analyzed for TCL Pesticides and PCBs. None of these compounds were detected in these Phase 2 samples.

2.15.4 Metals in Surface Water

The HHAWQC screening of Phases 1A and 1B surface water results indicated that arsenic, iron, and manganese were detected in at least one surface water sample above their respective HHAWQC values. Iron was most frequently detected (28 of the 35 surface water samples) above the HHAWQC (300 µg/L) during Phases 1A and 1B. Iron concentrations exceeding the HHAWQC ranged from 330 µg/L in SW6 to 2140 µg/L in SW49. Manganese was detected above the HHAWQC in six surface water samples collected during Phase 1 of the RI. Manganese concentrations exceeding the HHAWQC (50 µg/L) ranged from 52.7 µg/L in SW13 to 192 µg/L in SW52. Arsenic was detected at concentrations exceeding the HHAWQC (0.018 mg/kg) in four samples collected during Phase 1 of the RI. The arsenic concentrations above the HHAWQC ranged from 2.7 µg/L in SW13 to 3.7 µg/L in SW23. Arsenic, iron and manganese have been detected in soil and sediment throughout the investigation area in similar concentrations.

Comparison of Phase 1 surface water data against ecological screening values indicated the presence of elevated copper, nickel and zinc concentrations at three locations within the intermittent stream (SW6, SW10, and SW11). Copper concentrations in these samples ranged from 4.2 µg/L to 13.9 µg/L, exceeding the South Carolina Water Quality Criteria (SCWQC) of 1.26 µg/L. Nickel concentrations in these samples ranged from 11.3 to 71.6 µg/L, exceeding the SCWQC of 7.18 µg/L. Zinc concentrations in these samples ranged from 105 µg/L to 178 µg/L, exceeding the SCWQC of 16.5 µg/L. Arsenic, iron and manganese were detected above their HHAWQC in background surface water samples collected during the RI. Arsenic concentrations ranged from 2.6 µg/L in BKSW4 to 3.4 µg/L in BKSW2. Iron concentrations in background surface water samples range from 186 µg/L in the BKSW1 to 25730 µg/L in BKSW4. Manganese

concentrations in background surface water samples range from 16.7 µg/L in BKSW1 to 189 µg/L in BKSW2.

During Phase 2, seven sets of surface water samples were collected from the Site. Three samples (BKSW2 through BKSW4) were collected from previously unsampled background locations to support the macroinvertebrate community sampling program. Four samples (SW11, SW23, SW27, and SW51) were collected from previously sampled locations as part of the surface water/groundwater interaction investigation. During Phase 2B, surface water samples were again collected from SW11 and SW23 as part of the surface water/groundwater interaction investigation

A comparison of Phase 2 and Phase 2B surface water sample results indicate that only the compounds arsenic and thallium were detected above their respective HHAWQC values. Arsenic concentrations ranged from 2.6 µg/L in BKSW3 to 3.4 µg/L in BKSW2. Thallium was detected in one surface water sample above its HHAWQC, BKSW3 at 5.6 µg/L).

A comparison of Phase 2 surface water sample analytical results indicate that nickel and zinc concentrations at SW11, SW23, SW27, SW51 also exceed their respective SCWQCs. Phase 2 surface water nickel concentrations exceeding the SCWQC ranged from 59.3 µg/L at SW27 to 97.7 µg/L at SW11. Phase 2 surface water zinc concentrations exceeding the SCWQC ranged from 68.4 µg/L (dissolved zinc) at SW51 to 97.6 µg/L at SW23. Nickel and zinc concentrations also exceed their respective SCWQCs in the Phase 2B surface water samples from SW11 and SW23. Phase 2B surface water nickel concentrations exceeding the SCWQC ranged from 68.4 µg/L (dissolved nickel) at SW23 to 71.4 µg/L at SW11. Phase 2B surface water zinc concentrations exceeding the SCWQC ranged from 211 µg/L (dissolved zinc) at SW23 to 239 µg/L at SW11.

2.16 SOURCE AREAS

A variety of samples were collected from the waste within the potential source areas. A removal action conducted in January and February of 2005 removed three source areas, the Imhoff Tank, the Sludge Drying Bed, the Trickling Filter and their respective contents. This section briefly summarizes the results for waste samples collected from the remaining source areas, specifically the area surrounding the former Imhoff tank and the Equalization lagoon.

2.16.1 Area Surrounding Former Imhoff Structures

During various phases of the RI, surface soil samples, subsurface soil samples, and groundwater samples were collected from locations adjacent to and surrounding the Imhoff system structures. Surface and subsurface soil samples were collected from soil borings (TMW32 and TMW33) and test pits (TP17 through TP21). The locations of these sample points are listed below:

TMW32 southeast side of the Sludge drying bed

TMW33 northwest side of the Sludge drying bed

TP17 northeast corner of the Imhoff tank

TP18 southeast corner of Imhoff tank

TP19 southeast corner of Trickle filter

TP20 southeast side of Trickle filter

TP21 southwest corner of Trickle filter

Subsurface soil samples were also collected from beneath the sludge drying bed at locations SB8, SB9, SB10, and SB13. Subsurface soil samples obtained from these test pits were collected from beneath the edges of the Imhoff structures. In addition, surface soil samples were collected from locations SS1, SS2, SS3, SS26, and SS27 which are located adjacent to the southern and eastern sides of the trickle filter.

Groundwater samples were collected from wells TMW32, TMW33, MW12, MW12D (northwest side of Imhoff tank), and MW13 (east of Imhoff system structures). Data from the permanent wells has been used to define the extent of TCE and CT concentrations in groundwater in the Imhoff system area. Analytical data from these sample points do not indicate that the contents of Imhoff system structures have leaked in the subsurface creating a separate source of contaminants in this area.

2.16.2 Former Equalization Lagoon

A variety of investigative activities were performed in and around the equalization lagoon during the RI. Four test pits (TP1 through TP4) were excavated through the lagoon. One well boring (TMW12) was installed through the boring to allow collection of a groundwater sample from the water table. Surface soil, waste samples, and native subsurface soil samples were collected from each of the test pits and TMW12 during Phase 1A. During Phase 1B, two MWs (MW5 and MW5D) were installed immediately adjacent to the lagoon. During Phase 2, two additional MWs (MW5D2 and MW5D3) were installed adjacent to the lagoon. Two native subsurface soil samples (SB18 and SB19) were collected from beneath the lagoon.

Review of the results for soil samples collected from the test pits indicate that only the metals arsenic, chromium, and iron were detected above their respective USEPA Region 9 residential PRGs in soil samples. Arsenic was detected at or above its residential (0.39 mg/kg) and industrial (1.5 mg/kg) PRGs in each of the surface and subsurface soil samples collected from the test pits, ranging in concentrations from 1.5 mg/kg in 01TP3AA to 10.6 mg/kg in 01TP4CA. Chromium was detected above its residential and industrial PRGs for Cr(VI) in one subsurface soil sample collected from beneath the equalization lagoon, 01TP1CAA at 76.7 mg/kg. Chromium was also detected in the lagoon waste samples at concentrations ranging from 2380 mg/kg in 01TP4BA to 01TP2BA at 8550 mg/kg. Chromium was also detected in the waste sample

01TMWS12BA at 6710 mg/kg. During installation of the permanent monitoring wells the Color Tec screening methodology was used to screen soil and groundwater samples collected from various intervals for evidence of impact by chlorinated compounds. This was performed to determine if the former lagoon is a source for the TCE found in groundwater beneath the Site.

2.17 Investigative Activities at Dixie-Narco Plant

2.17.1 Geophysical Survey

A variety of investigative activities have been performed at the Dixie-Narco Plant. A geophysical survey was performed over the parking area at the eastern end of the Dixie-Narco plant during Phase 1A. The geophysical survey located the stormwater sewer line that runs to the southeast beneath the rear parking area; portions of two active sanitary sewer lines that run north-south beneath the rear parking area; the eastern boundary of the former equalization lagoon; and an anomalous area in the north central portion of the rear parking area.

Based on the results of the geophysical surveys, during Phase 1B, a large anomaly located in the north central portion of the rear parking area was investigated. Three trenches (TP22 through TP24) were excavated through the anomaly. Each of the three trenches encountered a variety of debris buried between one foot and four feet below land surface. The debris included plastic sheeting, wood fragments, cinder block fragments, corrugated metal sheeting, steel banding and other miscellaneous material. A section of corrugated concrete pipe approximately 6 inches in diameter oriented northeast to southwest was also encountered in TP23 and TP24. This piping is believed to be a section of the remnant sewer pipe system installed when the facility was originally used as a mobile home community in the 1950's.

None of the debris encountered in the trenches exhibited any visual characteristic that might be considered a potential threat to the subsurface soil quality.
A subsurface soil sample was collected from each test pit. These soil samples were analyzed for the TCL/TAL suite of parameters. Arsenic is the only parameter detected in these three samples above its industrial and residential PRGs. The arsenic concentrations in these samples range from 4.1 in 02TP23AA to 5.4 mg/kg in 01TP22AA.

2.17.2 Former Forklift Repair Shop

During Phase 2 of the RI, several temporary wells were installed on the northwest side of the Dixie-Narco plant (TMW56 through TMW66). Groundwater sample analytical results for the samples from these wells indicated a "hot spot" for TCE in groundwater in this area. The groundwater sample collected from TMW56 contained the highest TCE concentration detected at the Site - 410 µg/L. Review of historical information from the plant revealed that a forklift repair shop was located in the area in which the TCE hot spot was located.

As a result of the temporary well program, four monitoring wells were installed in this area. MW26 was installed to the northwest of the plant as an additional background well. Three wells (MW27, MW27D, and MW27D2) were also installed in the vicinity of the forklift repair shop, nearest the location for TMW56.

2.18 Summary of Site Risks

This section includes a summary of the human health and ecological risk assessment activities performed at the AHA Site. The complete versions of the risk assessments are contained in the Administrative Record.

2.18.1 Human Health Risk Assessment

This section summarizes the baseline human health risk assessment (HHRA) for the AHA Site. The baseline risk assessment estimates what risks the site poses if no action were taken. It provides the basis for taking action and identifies the contaminants and exposure pathways that need to be addressed by the remedial action. This section of the ROD summarizes the results of the baseline risk assessment for this site.

The baseline HHRA has been conducted in accordance with the four-step paradigm for human health risk assessments developed by U.S. EPA (1989a); these steps are:

- Data Evaluation and Hazard Identification
- Toxicity Assessment
- Exposure Assessment
- Risk Characterization

Each of these four steps is summarized below, followed by a summary and conclusions.

2.18.1.1 Data Evaluation and Hazard Identification

The purpose of the data evaluation and hazard identification process is two-fold: 1) to evaluate the nature and extent of release of chemicals present at the site; and 2) to select a subset of these chemicals identified as COPCs for quantitative evaluation in the HHRA. This step of the risk assessment involves compiling and summarizing the analytical data, and selecting COPCs based on a series of screening steps.

The document entitled, "*Technical Memorandum: Human Health Chemicals of Potential Concern Screening*" (ENSR, 2004a) presents the COPCs identified for the Site. The human health COPCs were selected based on a series of screening steps that consisted of comparison to site-specific background values, essential nutrient status, and comparison to toxicity (health-based) benchmarks. 34 site-specific COPCs were included in the quantitative risk assessment.

2.18.1.2 Toxicity Assessment

The purpose of the toxicity assessment is to identify the types of adverse health effects a chemical may potentially cause. This assessment is followed by a dose response assessment to define the relationship between the dose of a chemical and the likelihood or magnitude of an adverse effect (response) (U.S. EPA, 1989a). Adverse effects are classified by U.S. EPA as potentially carcinogenic or noncarcinogenic (i.e., potential affects other than cancer). Dose-response values were used to evaluate oral, dermal, and inhalation exposures to COCs.

Dose-response values were obtained from several sources within U.S. EPA, including the Integrated Risk Information System (IRIS) and the Provisional Peer Reviewed Toxicity Values (PPRTVs), as well as California EPA (CalEPA). The U.S. EPA's guidance regarding the hierarchy of human health toxicity values in risk assessment was followed, along with specific guidance provided by Region 4. In the case of trichloroethene (TCE), both the upper end of U.S. EPA's range of provisional draft cancer slope factors (0.4 per mg/kg-day for both oral and inhalation exposures) and the CalEPA's cancer slope factors (0.013 per mg/kg-day (oral exposure) and 0.007 per mg/kg-day (inhalation exposure)) were used to evaluate potential cancer risk posed by TCE, per Region 4 recommendation. TCE is a COC only in groundwater.

2.18.1.3 Exposure Assessment

The purpose of the exposure assessment is to predict the magnitude and frequency of potential human exposure to each of the COCs retained for quantitative evaluation in the HHRA. The first step in the exposure assessment process is the characterization of the setting of the Site and surrounding area. Current and potential future Site uses and potential receptors (i.e., people who may contact the impacted environmental media of interest) are identified based on information on land use. Potential exposure scenarios identifying appropriate environmental media and exposure pathways for current and potential future site uses and receptors are then developed. Those potential exposure pathways for which COCs are identified and are judged to be complete are evaluated quantitatively in the risk assessment.

The Dixie-Narco plant has been used for industrial purposes for many years and this facility is expected to remain industrial well into the reasonable foreseeable future. Therefore, current and future on-site receptors have been identified based on an industrial land use scenario for the manufacturing facility (identified as the on-site area west of County Road 65) and the triangular parcel owned by Dixie-Narco located immediately southeast of County Road 65. Groundwater at the plant is not currently used for drinking water or any other purpose. All water at the plant is supplied by the Town of Williston's municipal drinking water system. However, use of on-site groundwater as drinking water is included in the future use scenario for the on-site worker receptors.

The Imhoff system is located on an undeveloped lot (the area east of County Road 65) now owned by Dixie-Narco. Future residential use of this lot is not considered likely; however, both industrial and future residential use of this parcel (excluding formally delineated wetland) was evaluated in the HHRA. The Imhoff system (Upland area and Wetland area), Spur Branch (consisting of the intermittent and perennial stream stretching from the Imhoff wetland area down to Willis Millpond), Willis Millpond, and downstream of Willis Millpond are identified as off-site exposure areas in the HHRA. For the purposes of the HHRA, the manufacturing plant area and the adjacent triangular parcel owned by Dixie-Narco are within the plant property boundary and are termed "on-site." All other areas included in the RI investigation are outside the plant property boundary and are termed "off-site" in the HHRA.

Figure RD-4 presents a conceptual site model identifying sources, migration pathways, potential human receptors and potential exposure pathways for the Site. The potentially complete exposure pathways evaluated in the risk assessment are identified in this figure. As shown in Figure RD-4, potentially complete exposure pathways are identified for the following receptors at the Site:

- On-site outdoor worker
- On-site indoor worker
- Construction/utility worker
- Trespasser
- Wader
- Off-site resident

Figure RD-4 of the ROD presents a matrix of receptors and exposure scenarios (pathways and areas) evaluated in the HHRA. For each receptor/exposure pathway combination, both Reasonable Maximum Exposure (RME) and Central Tendency Exposure (CTE) assumptions were evaluated.

The calculation of potential exposure point concentrations (EPCs) and exposure doses was performed in accordance with current U.S. EPA. The most recent U.S. EPA guidance entitled, "Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites" (U.S. EPA, 2002) was used to calculate EPCs for the RME scenarios. EPCs for certain exposure pathways were modeled from groundwater, soil vapor, and soil data (e.g., indoor air, fugitive dust) as proposed in the work plan using EPA-approved models/methods.

2.18.1.4 Risk Characterization

Risk characterization is the process in which estimated human exposure doses (derived in the Exposure Assessment) are integrated with dose-response values (presented in the Toxicity Assessment) to generate estimates of risk. Two general types of health risk are characterized for each potential exposure pathway considered: potential noncarcinogenic

risk and potential carcinogenic risk. Given the nature of the risk assessment process (i.e., conservative exposure assumptions, exposure point concentrations, and toxicity values), predicted risk estimates are considered to be upper-bound.

Potential cancer risks for each receptor were compared to the low end (point of departure) of the U.S. EPA target risk range (i.e., 1×10^{-6}) for chemical of concern (COC) identification. Risk management decisions are made within the risk range of 1×10^{-6} to 1×10^{-4}. For noncarcinogens, a target cumulative site Hazard Index (HI) of one (for chemicals that act on the same target organ) (U.S. EPA, 1991a, U.S EPA, 2000) is used for COC identification. In addition, any COPC with a concentration in a monitoring well or residential well above the health based MCL is also identified as a COC for groundwater.

Two sets of TCE cancer risks were calculated, representing upper and lower bound risk estimates for this chemical. As shown in the RI/FS, potential RME and CTE cancer and noncancer risks for the current on-site outdoor worker and current on-site indoor worker, the future construction/utility worker, the current/future trespasser and the current/future wader are all below the U.S. EPA risk management benchmarks. Thus, based on the HHRA, no adverse risks are expected for these receptors.

Potential RME and CTE cancer and noncancer risks for the future on-site indoor and outdoor workers exceed the respective risk management benchmarks due to TCE, PCE, carbon tetrachloride and benzene in drinking water. However, as noted above, no one is currently drinking on-site groundwater. However, all groundwater in South Carolina is classified as GB (potable water) and thus should meet MCLs.

Potential RME and CTE cancer and noncancer risks for the hypothetical future off-site resident in the Imhoff Upland area exceed the respective risk management benchmarks due principally to TCE, PCE, and carbon tetrachloride in groundwater. However, as noted above, the Imhoff Upland area is currently undeveloped and future residential use of this parcel is considered unlikely.

For off-site groundwater, potential drinking water risks were predicted on a well-by-well basis (149 locations). Potential RME (and CTE) risks are above 1×10^{-6} and/or a hazard index of 1 (on a target organ basis) at the following monitoring well locations:

- MW11, MW11D, MW11D2, MW12, MW12D, MW13, MW14, MW15, MW15D, MW16, MW16D, MW17, MW18, MW19D, MW21D, MW29D2, MW29D2(58), MW29D2(98), MW30D, MW30D3, MW31, MW31D, MW32D, MW33D, MW36D, MW37D, MW38D, TMW3, TMW28, TMW29, TMW30, TMW31, TMW35, TMW36, TMW39, TMW40, TMW41, TMW51, TMW52, TMW53, TMW57, TMW58, TMW60, TMW61, TMW67, TMW68, TMW73, TMW74, TMW75, TMW80, TMW83, TMW84, TMW86, TMW87, TMW89, TMW90, and TMW92.

In addition to these off-site monitoring wells, potential risks for ten of the existing residential wells (RW1, RW5, RW7, RW13, RW20, RW25, RW38, RW49, RW54, and RW57) exceed the $1x10^{-6}$ target risk level and two of the existing residential wells (RW54 and RW57) exceed the target HI of 1. The target risk level exceedance in six of the residential wells is due principally to TCE (RW1, RW5, RW7, RW20, RW49, and RW57). The target risk level exceedance for RW25 is due to PCE. It should be noted that the concentrations of TCE and PCE detected in each of these wells, with the exception of RW57, are below their respective MCLs.

For RW13, the unacceptable potential cancer risk is due to benzo(a)pyrene and indeno(1,2,3-cd)pyrene. Benzo(a)pyrene and indeno(1,2,3-cd)pyrene were detected in

RW13 in November, 2001, however, when RW13 was resampled in April, 2002, neither chemical was detected. The presence of both PAH compounds in the November 2001 sample is attributed to inadvertent collection of soil particulates to which the PAH compounds were bound. Therefore, based on the more recent groundwater sampling data, potential exposure to constituents detected in groundwater at RW13 is not considered to pose unacceptable potential carcinogenic risk, and neither benzo(a)pyrene nor indeno(1,2,3-cd)pyrene is identified as a COC in groundwater at RW13.
For RW38, the unacceptable potential cancer risk is due principally to benzo(a)pyrene. However, the potential carcinogenic risk posed by benzo(a)pyrene alone is below the $1x10^{-6}$ target risk level. Therefore, potential exposure to constituents detected in groundwater at RW38 is not considered to pose unacceptable carcinogenic risk. Furthermore, as with RW13, the presence of benzo(a)pyrene in groundwater at RW38 is attributed to inadvertent collection of soil particulates during sampling. Therefore, benzo(a)pyrene is not identified as a COC in groundwater at RW38.

For RW54, the unacceptable potential cancer risk is due to arsenic and the unacceptable potential hazard index is due to iron. However, RW54 is located southwest and cross-gradient of the Site. Hence, it is unlikely that RW54 has been affected by groundwater from the Site. It should also be noted that the concentration of arsenic detected in RW54 is below the MCL.

Based on the risk characterization results (i.e. total risk exceeds risk level of 1×10^{-6} and/or hazard index of 1), eight chemicals were identified as COCs in groundwater (benzene, carbon tetrachloride, dichloromethane, 1,1-dichloroethene, tetrachloroethene, and trichloroethylene, mercury and nickel). In accordance with U.S. EPA guidance (2000a), remedial goal options (RGOs) were derived for the COCs. RGOs were calculated based on target cancer risk levels of 1×10^{-6}, 1×10^{-5} and 1×10^{-4}, and target hazard quotients of 0.1, 1 and 3. Where available, Applicable or Relevant and Appropriate Requirements (ARARs) are also presented for the COCs, which consist of federal and state drinking water standards, referred to as Maximum Contaminant Levels (MCLs).

2.18.1.5 Summary and Conclusions

Based on the results of the HHRA, the only medium/exposure pathway that poses a potentially unacceptable potential risk to human health is consumption of on-site groundwater by a future worker and consumption of off-site groundwater by a resident at off-site well locations. Eight of these locations are existing residential wells (RW1, RW5, RW7, RW20, RW25, RW49, RW54, and RW57). At RW1, RW5, RW7, RW20, and RW49, where TCE is the only COC, detected TCE concentrations are all well below the TCE MCL. Further, when the CalEPA cancer toxicity value for TCE is used to calculate potential drinking water risk, the total risk at each of these wells falls below the target risk level of 1×10^{-6}. Therefore, the inclusion of these wells in the risk characterization summary and identification of TCE as a COC at these wells is conservative. At RW25, where PCE is the only COC, the detected PCE concentration is well below the PCE MCL. At RW57, where TCE was detected in November 2003 at a concentration above its MCL, a water filtration system for the water supply at this property was installed in March 2004. No one is currently residing at this location. As previously noted, RW54, where arsenic and iron are elevated is southwest and cross-gradient of the Site, making it unlikely that this well is affected by groundwater from the Site. It should also be noted that iron is an essential nutrient and there is considerable uncertainty in the oral toxicity value provided by USEPA and used in the HHRA. It is a provisional value with a medium level of confidence assigned by the agency.

In summary, unacceptable on-site drinking water potential risks for workers (assuming future use of on-site groundwater) are due principally to TCE, and to a lesser extent, benzene, carbon tetrachloride and tetrachloroethylene (PCE). Unacceptable potential drinking water risks at off-site monitoring well locations (non-residential wells) are due principally to TCE, and to a lesser extent, carbon tetrachloride, tetrachloroethylene, arsenic, iron, manganese, mercury, and nickel. It should be noted that no one is currently drinking groundwater at any of these off-site monitoring well locations. For six of the seven residential wells where potential unacceptable risks were predicted due to TCE and/or PCE (RW1, RW5, RW7, RW20, RW25, and RW49), detected concentrations of these two chemicals are all below MCLs. For RW57, where TCE has been detected above its MCL, groundwater at that location is not currently used for drinking water and the residence was supplied with a water filtration system. For the residential well RW54, its location southwest of the Site and cross-gradient of site groundwater flow, and the fact that the two COCs posing potentially unacceptable risk are naturally occurring and not likely to be site-related (arsenic and iron), make it unlikely that RW54 has been affected by the Site.

2.18.2 Ecological Risk Assessment Summary

This section summarizes the Baseline Ecological Risk Assessment (BERA) for the AHA Site. A complete report describing the risk assessment methodology and results is provided in a separate stand-alone document (the BERA Report). The AHA Ecological Risk Assessment (ERA) program provides an evaluation of the potential risks to

ecological receptors posed by COPCs in surficial environmental media (surface soils, sediment, and surface water) at the Site. The ERA process at the AHA Site was conducted in several tiers or phases. These tiers are described below:

- The first portion of the ecological evaluation consisted of a Preliminary Screening Level ERA (PERA) presented in the Phase I RI Work Plan (ENSR, 2002). The PERA concluded that there was a potential for risk of harm to ecological receptors from exposure to site-related chemicals in sediments (and potentially surface water) and additional evaluation was proposed.
- The second portion of the ecological evaluation consisted of a Screening Level ERA (SERA) based on data collected during the Phase I RI field program (ENSR, 2003). The results of the SERA indicated that a conclusion of "no significant risk" could not be reached for ecological receptors potentially exposed to hydric soils and sediment at the AHA Site.
- The BERA Problem Formulation Statement (PFS) was prepared to help establish the goals, breadth, and focus of the BERA. The PFS indicated that additional evaluation of selected metals and pesticides in hydric soil and sediment was warranted in three areas: the Imhoff System wetland, the Intermittent Spur Branch, and the Willis Millpond.
- A BERA Work Plan was submitted concurrently with the SERA/PFS (ENSR, 2003b). The BERA Work Plan serves as the basis for the BERA Report.

The BERA Report is based on the completed Phase II RI field program. The BERA refines the screening-level risk evaluation, and focuses on potentially complete exposure pathways of potential concern. The BERA provides estimates of potential ecological risks through a weight-of-evidence (WOE) approach. The BERA at the AHA Site was conducted in accordance with relevant state and federal guidance, including the following:

- Framework for Ecological Risk Assessment (U.S. EPA, 1992);
- Region 4 Ecological Risk Assessment Guidance (U.S. EPA, 2001);
- Intermittent "ECO Update" Bulletins of U.S. EPA;
- Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (Interim Final) (U.S. EPA, 1997); and
- Guidelines for Ecological Risk Assessment (U.S. EPA, 1998).

The principal components of the BERA include:

- **Problem Formulation**: In this phase, the objectives of the ERA are defined, and a plan for characterizing and analyzing risks is determined.
- **Risk Analysis**: Risk analysis is directed by the problem formulation. During this phase of work, data are evaluated to characterize potential ecological exposures and effects.

- **Risk Characterization**: During risk characterization, exposure and stressor response profiles are integrated through risk estimation. Risk characterization also includes a summary of uncertainties, strengths, and weaknesses associated with the risk assessment.

2.18.2.1 Summary of BERA Field Effort

Figure RD-5 depicts a conceptual site model for the Baseline Ecological Risk Assessment. The BERA field program was completed in August and September of 2003, and included biological, chemical, and ecotoxicological sampling and analysis activities from the following three exposure areas with a potential for ecological risks:

1. Imhoff System wetland,
2. Intermittent portions of Spur Branch, and
3. Willis Millpond.

In order to provide information for the BERA WOE risk characterization, a number of analytical chemistry sampling locations in these three areas were co-located with toxicity testing and/or macroinvertebrate community analysis stations. The BERA report focuses primarily on these stations with synoptically collected (in time and space) chemical, biological, and toxicological data. Data evaluated in the BERA include:

- Analytical chemistry results (selected inorganic constituents, pesticides, simultaneously extractable metals (SEM), acid volatile sulfides (AVS), and total organic carbon (TOC) from nine surficial sediment samples collected from Site locations and three sediment samples collected from regional background sampling locations.
- Analytical chemistry results (selected inorganic constituents, pesticides, SEM, AVS, and TOC) from ten hydric surface soil samples collected from Site locations and two regional background sampling locations.
- Laboratory toxicity testing results from the nine on-Site and three background sediment sampling locations. These species included two invertebrates: a midge (*Chironomus tentans*) and an amphipod (*Hyalella azteca*).
- Laboratory toxicity testing results from the ten on-Site and two background hydric soil sampling locations. Test species included a plant [Japanese millet (*Echinochloa crusgalli*)] and the earthworm (*Eisenia foetida*).
- Site-specific macroinvertebrate community survey data from six of the nine on-site sediment analytical chemistry and sediment toxicity test sampling stations, as well as at three background stations

2.18.2.2 Benthic Receptors

Four different measurement endpoints were considered in the benthic macroinvertebrate risk characterization. These measurement endpoints were designed to evaluate the sustainability of a healthy and well-balanced benthic invertebrate community in the Imhoff System wetland, Spur Branch intermittent stream and Willis Millpond which is

typical of comparable South Carolina streams and ponds with similar structure, morphology, and hydrology. These measurement endpoints are discussed below:

- Sediment benchmark screening indicates benthic receptors are at risk due to exposure to inorganic COPCs in the Imhoff System wetland (stations SD55 and SD5), portions of Spur Branch closest to the Imhoff System (stations SD6 and SD7), and Willis Millpond (stations SD58, SD59, and SD60).
- The SEM/AVS and $(\Sigma SEM\text{-}AVS)/f_{oc}$ evaluations indicates that divalent metals are bioavailable, and potentially toxic, in one Imhoff System wetland station (SD5) and three of the four Spur Branch stations (SD6, SD7, and SD16). Metals in Willis Millpond samples were not predicted to be bioavailable to benthic receptors.
- The results of toxicity testing with two species of benthic invertebrates suggest that sediments in the Imhoff System wetland and the upstream intermittent portions of the Spur Branch are negatively impacting the benthic receptor community. Willis Millpond samples did not exhibit significant toxicity when compared to the reference location.
- The macroinvertebrate community survey data suggested that water quality in the more downstream portions of Spur Branch (SD7, SD16, and SD22) was better than in many other stations. However, this may be due to the presence of flowing water as opposed to constituent concentrations. The benthic communities at stations SD5, SD55, and SD6 appear to be adversely impacted. Since the macroinvertebrate evaluation methodology was developed for flowing streams, it may be difficult to reach conclusions for the more palustrine swampy habitat present at the AHA Site.

In conclusion, a potential for ecological risks to the benthic macroinvertebrate community exists in the upper portions of the intermittent Spur Branch and the Imhoff System wetland.

2.18.2.3 Wetland Invertebrates

Three different measurement endpoints were considered in the wetland invertebrate risk characterization. These measurement endpoints were designed to evaluate the sustainability of a wetland invertebrate community which reflects the available habitat in the wetland areas near the Imhoff System and Willis Millpond and can serve as a forage base for higher trophic level receptors. These measurement endpoints are discussed below:

- Hydric soil benchmark screening indicates wetland invertebrate receptors at risk due to exposure to inorganic COCs in the Imhoff System wetland (stations SS12, SS1, SS14, SS15, SS30, SS5, and SS28) and Willis Millpond (stations SS58, SS57, and to a lesser extent SS55). Wetland invertebrates are also at risk due to exposure to pesticides in hydric soil in the same stations. There is some uncertainty in this benchmark evaluation because earthworm

toxicity screening values were not available for all COCs and alternative soil screening values were used in these cases.

- The SEM/AVS evaluation indicates that divalent metals are bioavailable, and potentially toxic, in all six Imhoff System wetland stations and one of the three Willis Millpond stations (SS55). The analysis of the divalent cationic metals data evaluated through the $(\Sigma SEM\text{-}AVS)/f_{oc}$ approach suggests that divalent metals at stations SS5 and SS30 are potentially less bioavailable than would be predicted on the basis of total concentration alone, and pose significantly less risk to benthic ecological receptors than predicted using simple comparisons of total concentrations to metals benchmarks. Stations SS1, SS12, SS14, SS15, SS28, and SS55 fall into the intermediate category in the $(\Sigma SEM\text{-}AVS)/f_{oc}$ evaluation where prediction of effects are uncertain.
- The results of the earthworm toxicity testing indicate that hydric soils in the Imhoff System wetland and the Willis Millpond perimeter wetland are not having a negative impact on the wetland invertebrate communities. The results of the site-specific toxicity testing are considered to be a better estimator of potential risk at the AHA Site than the more generic benchmark screening.

In conclusion, an evaluation of the screening data demonstrates a strong potential for ecological risks. The site-specific toxicity testing to date suggests that there does not appear to be significant potential for risk to the wetland invertebrate community in the Imhoff System wetland (icluding the intermittent stream) or the Willis Millpond perimeter wetland. Therefore, additional toxicity testing will be performed in years 1 and 5 of the Remedial Action.

2.18.2.4 Wetland Plants

Two different measurement endpoints were considered in the wetland plant risk characterization. These measurement endpoints were designed to evaluate the sustainability of a wetland plant community which reflects the available habitat in the palustrine wetland areas near the Imhoff System and Willis Millpond and can serve as a forage base for higher trophic level receptors. These measurement endpoints are discussed below:

- Hydric soil benchmark screening indicates wetland plant receptors are at risk due to exposure to inorganic COPCs in the Imhoff System wetland (stations SS12, SS1, SS14, SS15, SS30, SS5, and SS28) and Willis Millpond perimeter wetland (stations SS58, SS57, and to a lesser extent SS55). Wetland plants are also be at risk due to exposure to pesticides in hydric soil in the same stations. There is some uncertainty in this benchmark evaluation because phytotoxicity screening values were not available for all COPCs and alternative soil screening values were used in these cases.

- The results of the plant toxicity testing indicate that hydric soils in the Imhoff System wetland are having a negative impact on the wetland plant community. Stations within the Imhoff System wetland exhibited lethal and/or sub-lethal effects. Stations within the Willis Millpond perimeter wetland were impacted to a lesser degree with reductions in root growth, but no impacts on survival or shoot growth. The results of the site-specific toxicity testing are considered to be a better predictor of risk at the AHA Site than the more generic benchmark screening.

In conclusion, there does not appear to be significant potential for risk to the wetland plant community in the Willis Millpond perimeter wetland. However, the potential for risk to the wetland plant community exists within the Imhoff System wetland.

2.18.2.5 BERA Weight of Evidence (WOE) Conclusions

Based on interpretation of a weight of evidence analyses and best professional judgment, the following conclusions can be made regarding the overall potential risk within each of the three AHA sampling areas.

- The potential for ecological risks to wetland plants from exposure to site-related COPCs in hydric soils exists in the Imhoff System wetland exposure area.
- The potential for ecological risks to benthic macroinvertebrate receptors from exposure to site-related COPCs in sediments exists in the most upstream portion of the intermittent Spur Branch, closest to the Imhoff System and including the stream portions located within the Imhoff System wetland.
- Although selected Willis Millpond COPCs exceed sediment and hydric soil benchmarks, the toxicity testing indicates minimal potential for ecological risk due to exposure to site-related COPCs in this exposure area.
- The results of the BERA indicate little potential ecological risks to wetland invertebrates.

Aquatic ecological receptors may be at risk from exposure to surface water concentrations of several inorganic compounds (e.g., copper, nickel, and zinc) which are present in excess of water quality standards and criteria in Spur Branch. It is presumed that remediation of the sediments and hydric soils in the source area will result in a substantial reduction in metals concentrations in the downstream receiving water column.

2.18.2.6 Response Action Outcomes/PRGs

Based on the results of the Ecological Risk Assessment, the complete ecological exposure pathways to the sediment/hydric soil in the Imhoff System wetland and intermittent portions of Spur Branch should be eliminated. COPCs in sediment and hydric soil include barium, beryllium, cadmium, copper, chromium, cobalt, iron, lead, mercury, nickel, selenium, vanadium, zinc, and potentially pesticides (although pesticides

are not site-related). There is the possibility that multiple stressors are contributing to observed hydric soil and sediment ecological risks at the AHA Site. Of these multiple stressors, based on the results of the BERA, the principle chemical stressors of potential concern include chromium, nickel, and zinc. These three inorganic constituents are: (1) co-located with one-another, as well as with the majority of other chemical stressors identified through the BERA process; (2) consistently present at elevated concentrations at sampling stations where toxicity was observed; and (3) were the primary potentially bioavailable constituents present in the SEM and AVS sampling.

Relative to the COCs in sediment and hydric soils, the following Remedial Action Objectives (RAOs) have been established:

- Implement measures that restore concentrations of COCs in hydric soils and sediment to clean up levels that are protective of direct contact by benthic macroinvertebrate ecological receptors; and
- Implement measures that restore concentrations of COCs in hydric soils to clean up levels that are protective of wetland plant communities.

In accordance with U.S. EPA policy and guidance [OSWER Directive 9285.7-17 (the 1994 Laws memo)] and the Ecological Risk Assessment Guidance for Superfund (U.S. EPA, 1997), ecological risk-based PRGs have been developed for presentation within the BERA. Several alternative ecological risk-based PRGs were evaluated in the BERA Report.

A range of PRGs for chromium, nickel, and zinc have been developed for sediment and hydric soil at the AHA Site. The selection of these compounds for PRG development is based on the understanding that elimination of complete ecological exposure pathways for selected COCs in sediment and hydric soil will address co-located COCs. The PRGs developed for the AHA Site reflect population and community level assessment endpoints, rather than individual organism endpoints.

2.19 Remedial Action Objectives

The Remedial Action Objectives (RAOs) are statements that identify the media and exposure pathways at the Site that will be addressed by the remedial actions to be protective of human health and environment. RAOs were developed for the exposure pathways considering the requirements of USEPA, SCDHEC, and the health based and ecological based risks of the constituents detected at the Site. The RAOs for the different environmental media at the Site are listed below:

2.19.1 Groundwater

The RAOs for groundwater are:

1. Implement measures to prevent the ingestion of groundwater containing COCs at levels above human health based MCLs,
2. Restore groundwater to MCLs for VOCs listed in Table 2-4.

2.19.2 Hydric Soils, Sediments, and Surface Water

The RAOs for the hydric soils, sediment, and surface water at the Site are:

1. Implement measures to remediate COCs in sediments in the intermittent Spur Branch to RGs that are protective of direct contact by benthic macroinvertebrate ecological receptors,
2. Implement measures to remediate COCs in hydric soils to RGs that are protective of wetland plant ecological receptors,
3. Implement measures to control the migration of COCs in the hydric soils and sediments into the intermittent Spur Branch in excess of clean up levels.
4. Implement measures to control sources of COCs to surface water in intermittent and perennial Spur Branch to RGs that are protective of organisms.

2.19.3 Former Equalization Lagoon

The RAOs for the equalization lagoon at the Site are as follows:

1. Implement measures to prevent the migration of COCs in the waste/sludge in the equalization lagoon into groundwater at levels that exceed their RGs,
2. Implement measures to prevent direct contact with COCs in the waste/sludge in the equalization lagoon at levels that could impact human health.

2.20 Description of Alternatives

The FS and the ROD developed and refined alternatives for each of three Site areas requiring remediation. The alternatives were developed for the former equalization lagoon (S alternatives), the groundwater (GW alternatives), and the discharge area and adjacent stream (sediments, hydric soil, surface water, the SHSSW alternatives). The alternatives and their respective costs are as follows:

2.20.1 Former Equalization Lagoon Alternatives (S)

2.20.1.1 S-1 No Action

Estimated Capital Costs-$0
Estimated Annual O & M Cost-$0
Estimated Present Worth Cost-$66,800.
Estimated Construction Timeframe-None
Estimated Time to Achieve RAOs-unknown

The no action alternative is included as a comparison baseline for the other alternatives. Since no action does not include any remediation, COCs in the former equalization

lagoon may persist at the Site. The cost of the five year review mandated by CERCLA is the only cost for this alternative. The present worth cost is for 6 five year reviews over a 30 year time period.

2.20.1.2 S-2 Excavation with Off-Site Disposal
Estimated Capital Costs-$55.200.
Estimated Annual O & M Cost-$0
Estimated Present Worth Cost-$122,000.
Estimated Construction Timeframe-1 year
Estimated Time to Achieve RAOs-30 years

Excavation with off-site disposal is designed to remove the source material in the lagoon that exceeds RGs. The former equalization lagoon is about 45 feet by 35 feet and is 8 feet beneath a current parking lot on the Dixie-Narco property. It is estimated that 470 cubic yards of material will be excavated above the sludge/waste, which is estimated to be approximately 240 cubic yards. Excavated sludge/waste and the soils will be stockpiled and characterized prior to disposal off-site. Excavated soils with COCs not exceeding RGs will be used to backfill the excavation. The excavation will be sampled from the bottom and sides of the excavation to ensure sufficient impacted material is removed.

2.20.1.3 S-3 Capping of Equalization Lagoon with Institutional Controls
Estimated Capital Costs-$19,560.
Estimated Annual O & M Cost-$2,400.
Estimated Present Worth Cost-$91,500.
Estimated Construction Timeframe-1 year
Estimated Time to Achieve RAO's-30 years

In this alternative, the sludge/waste is left in place in the lagoon and remains protected by the existing asphalt cap. To protect the integrity of the cap, deed restriction and periodic repair/maintenance will be necessary. The groundwater remedy's monitoring program will determine if leachates are reaching monitoring wells. Project life is assumed to be 30 years.

2.20.1.4 S-4 On-site stabilization/solidification and capping/Institutional Controls
Estimated Capital Costs-$92,380.
Estimated Annual O & M Cost-$1800.
Estimated Present Worth Cost-$219,200.
Estimated Construction Timeframe-1 year
Estimated Time to Achieve RAO's-30 years

This alternative consists of excavation of the sludge/waste and adding stabilizing agents (binder) such as cements, grouts, or fly ash. The material will be placed back into the excavation after the binder is added. Confirmatory sampling is a component of this remedy. To protect the integrity of the cap over the stabilized material, deed restriction and on-going repairs/maintenance will be required. Project life is 30 years.

2.20.2 Groundwater Alternatives (GW)

2.20.2.1 GW-1 No Action

Estimated Capital Costs-$0
Estimated Annual O & M Cost-$0
Estimated Present Worth Cost-$173,900.
Estimated Construction Timeframe-none
Estimated Time to Achieve RAOs-unknown

The no action alternative is developed as a comparative baseline for other alternatives. The only activity and costs are for a no action determination (including some baseline sampling) and for the required five year review, to be conducted 6 times over a 30 year time frame.

2.20.2.2 GW-2 Monitored Natural Attenuation (MNA)

Estimated Capital Costs-$0
Estimated Annual O & M Cost-Year 1-$205,600. Years 2-15 $99,700. Years 16-30 $52,700.
Estimated Present Worth Cost-$1,373,200.
Estimated Construction Timeframe-none
Estimated Time to Achieve RAOs-30 years

MNA involves natural attenuation of the impacted groundwater utilizing natural processes including adsorption, desorption, dilution, dispersion, volatilization, hydrolysis, and biodegradation. MNA remedies include these typical tasks: fate and transport modeling, collection of field biogeochemical indicators, and long term monitoring to document reductions in contaminant concentrations.

2.20.2.3 GW-3 Groundwater Pump & Treat with Monitored Natural Attenuation (MNA)

Estimated Capital Costs-$1,129,560.
Estimated Annual O & M Cost-$358,680.
Estimated Present Worth Cost-$5,496,700.
Estimated Construction Timeframe-1 year
Estimated Time to Achieve RAOs-30 years

This alternative involves collection of contaminated groundwater using appropriately spaced extraction wells and above ground treatment and disposal. Recovery wells would be installed at the Site at various locations and depths to allow extraction of groundwater containing dissolved chlorinated VOCs and contain the plume from migrating into off-site residential supply wells. At the Site, it is anticipated that 21 recovery wells would be installed in the TCE plume. In addition, 4 recovery wells would be installed in the leading edge of the TCE plume. Similarly, it is anticipated that 7 recovery wells would

be installed in the CT plume. The depth of the wells would range from 65 feet to 85 feet below ground level. Each well would be pumped at 2 to 3 gallons per minute.

The pumped groundwater would be treated using a shallow tray air stripper and liquid-phase granular activated carbon polishing unit prior to final discharge into a local storm sewer. The discharge would be required to comply with the requirements of the National Pollutant Discharge Elimination System (NPDES). MNA would be used to address and monitor COCs in groundwater in the remaining on-site and off-site areas. 30 monitoring wells, 32 recovery wells, and 10 residential wells would be sampled for the life of the project. The anticipated project life would be 30 years.

2.20.2.4 GW-4 Enhanced Reductive Dechlorination with MNA
Estimated Capital Costs-$939,360.
Estimated Annual O & M Cost- Year 1-$605,300 Years 2-3 $310,000.
Years 4-5 $99,700, Years 6-10 $51,400.
Estimated Present Worth Cost-$2,434,000.
Estimated Construction Timeframe-2 years
Estimated Time to Achieve RAOs-30 years

This alternative involves injection of biodegradable carbohydrate solution/electron donor substance such as vegetable oil, corn syrup, or sodium lactate via injection into the groundwater. These substances would create in-situ anaerobic treatment zones, or in simple terms, allow the breakdown of the chlorinated VOC contaminants into harmless compounds. It is estimated that 160 injection points/wells will be necessary in the TCE plume, and 90 injection points/wells in the CT plume.

A field-scale pilot study will be required to establish the design parameters. The number, placement, and exact locations of the injection points would then be established along with the schedule and amount of carbohydrate solutions to be injected. The MNA component is proposed to sample 30 monitoring wells and 10 residential wells with varying schedules of sampling.

A supplemental technology to ERD which may also be utilized is nanoscale bimetallic iron. This technology would help to aid in the dechlorination should subsurface conditions inhibit the progress of the ERD.

2.20.3 Sediment, Hydric Soil, & Surface Water Alternatives for Discharge Area, Wetlands and Stream (SHSSW)

2.20.3.1 SHSSW-1 No Action
Estimated Capital Costs-$0
Estimated Annual O & M Cost-$0
Estimated Present Worth Cost-$140,600.
Estimated Construction Timeframe-none
Estimated Time to Achieve RAOs-unknown

The no action alternative is included as a comparative baseline for other alternatives. The only two cost components are for the no action determination, including some baseline sampling, and for conducting 6 five year reviews over the 30 year time frame.

2.20.3.2 SHSSW-2 Removal and Off-site Disposal of Sediment and Hydric Soils and Monitored Natural Attenuation (MNA) of Downstream Surface Water and Sediments
Estimated Capital Costs-$1,582,560.
Estimated Annual O & M Cost-Years 1-5-$106,760. Years 6-30-$40,100.
Estimated Present Worth Cost-$2,591,000.
Estimated Construction Timeframe-2 years
Estimated Time to Achieve RAOs-30 years

The excavation with off-site disposal of sediments and hydric soils with MNA is designed to address the impacted shallow and subsurface hydric soils and sediments that represent significant risks. The SHSSW-2 alternative would excavate or dredge impacted soils and sediments with mechanical equipment in Wetland Cover Types A, B, and C. Excavation would be followed by confirmation sampling to insure all impacted soils are removed. It is estimated that over 3,050 cubic yards of hydric soil would be excavated along with 1800 feet of stream bed sediments yielding an additional 200 cubic yards.

The removed material may require additional dewatering before disposal in a permitted Subtitle D facility. MNA, including a focused confirmatory toxicity testing program at 38 sample locations has been included during year 1 and year 5 of the sampling program for Spur Branch between Charleston Street and Willis Millpond. After removal and disposal, the excavated wetlands would be backfilled with similar soils and graded appropriately, then restored in accordance with State and Federal requirements.

2.20.3.3 SHSSW-3 Capping of Hydric Soils in Imhoff Wetlands, Limited Removal of Sediments and Hydric Soils, and Monitoring Natural Attenuation of Downstream Surface Water
Estimated Capital Costs-$1,455,450.
Estimated Annual O & M Cost-Years 1-5 $109,460. Years 6-30 $38,200.
Estimated Present Worth Cost-$2,665,600.
Estimated Construction Timeframe-2 years
Estimated Time to Achieve RAOs-30 years

This alternative is designed to address the impacted shallow and subsurface hydric soils and sediments that represent significant risks. SHSSW-3 differs from SHSSW-2 in that impacted hydric soils and sediments would have limited excavation and instead be capped in place. It is estimated that 1300 cubic yards of hydric soil would be capped, 1750 cubic yards of hydric soil would be removed, and 200 cubic yards of sediment would also be removed. The MNA of the surface water would be monitored with a program identical to that proposed for SHSSW-2. The project life is 30 years.

2.21 Applicable or Relevant and Appropriate Requirements

In this section, ARARs are summarized and discussed. ARARs are used to determine the appropriate extent of site cleanup, to scope and formulate remedial action alternatives, and to govern the implementation and operation of the selected action.

2.21.1 Defining ARARs

EPA's guidelines on ARARs (EPA, 1988) defines ARARs as follows:
Applicable requirements are those cleanup standards, standards of control, and other substantive environmental protection requirements, criteria, or limitations promulgated under federal or state law that specifically address a hazardous substance, pollutant, containment, remedial action, location, or any other circumstances at a CERCLA site. Applicability implies that the remediation or the circumstances at the site satisfy all of the jurisdictional prerequisites or requirements.

Relevant and appropriate requirements are those cleanup standards, standards of control, and other substantive environmental protection requirements, criteria, or limitations promulgated under federal or state law that, while not "applicable" to a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstance at a CERCLA site, address problems or situations sufficiently similar to those encountered at the CERCLA site that their use is well suited to the particular site.

The relevance and appropriateness of a requirement is judged by combining a number of factors including characteristics of the remedial action, the hazardous substances in question, or the physical circumstances of the site with those addressed in the requirement. The origin and objective of the requirement may aid in the determination of relevance and appropriateness. A requirement judged to be relevant and appropriate must be complied with to the same degree as if it were applicable. However, more discretion can be used in the determination. Only part of the requirement may be considered relevant and appropriate and the rest dismissed if judged not to be relevant and appropriate in a given case.

The final group of regulations considered by USEPA are "To Be Considered (TBC) Material." TBCs are non-promulgated advisories or guidance documents issued by federal or state governments. They do not have the status of ARARs but can be considered in determining the necessary level of cleanup for the protection of human health or the environment.

Three categories of ARARs and TBCs are identified by EPA (1988):

1) Chemical-specific ARARs are usually health- or risk-based numerical values or methodologies used to determine acceptable concentrations of chemicals that may be found in or discharged to the environment;